Site selection of urban wildlife sanctuaries for safeguarding indigenous biodiversity against increased predator pressures

Amin Rastandeh*, Daniel K. Brown, Maibritt Pedersen Zari

Faculty of Architecture & Design, Victoria University of Wellington, Wellington, New Zealand

ABSTRACT

Biodiversity loss in urban landscapes is a global challenge. Climate change is a major driving force behind biodiversity loss worldwide. Using Wellington, New Zealand as a research site, the aim of this research is to show how the most suitable patches of vegetation in urban landscapes can be identified, ranked, and prioritised as potential urban wildlife sanctuaries. This is in order to protect vulnerable indigenous fauna from some of the indirect impacts of climate change such as increased predator pressures and the spread of diseases among urban fauna caused by rising temperatures. A GIS-based multi-criteria analysis of spatial composition and configuration of patches of vegetation was undertaken with reference to eight factors affecting the quality of habitat patches and accordingly fauna behaviours in urban landscapes. Results show that Zealandia, the Wellington Botanic Garden, the Town Belt, and Otari-Wilton's Bush are respectively the most important urban sites for establishing pest-free urban wildlife sanctuaries in the study area. This research reveals that patch size should not be considered as the single most important factor for the site selection of urban wildlife sanctuaries because the collective importance of other factors may outweigh the significance of patch size as a single criterion. Lessons learned in the course of this research can be applied in similar cases in New Zealand or internationally in order to facilitate the process of site selection for the establishment of urban wildlife sanctuaries in highly fragmented urban landscapes suffering from rising temperatures and other climatic changes.

1. Introduction

1.1. The importance of biodiversity

The quality of ecosystem functions and services depends in part on the overall level and health of biodiversity (Balvanera et al., 2006; Cardinale et al., 2006; Hector and Bagchi, 2007; Duffy et al., 2007; Isbell et al., 2011; Hooper et al., 2012; Pasari et al., 2013; Tilman et al., 2014; Lefcheck and Duffy, 2015). In line with research on biodiversity in natural and semi-natural areas, urban biodiversity has received increasing attention in recent years (Farinha-Marques et al., 2011; Müller and Kamada, 2011; Elmqvist et al., 2013; Murgui and Hedblom, 2017). Biodiversity loss in the Southern Hemisphere in particular is a challenge influencing the quality and quantity of ecosystem services and thereby the quality of human life (Chambers et al., 2013; Jupiter et al., 2014; Urban, 2015; Taylor and Kumar, 2016; Rastandeh et al., 2017b).

1.2. New Zealand as a fragile biodiversity hotspot

New Zealand has been recognised as one of the most unique biodiversity hotspots on the planet (Myers et al., 2000; Olson et al., 2001). New Zealand has been separated from other landmasses for around 85 million years, since dinosaurs were widespread on the planet but before early mammals (i.e. egg-laying mammal species) were present (Fleming, 1975; Parkes and Murphy, 2003). As a consequence, the long-term evolution of New Zealand indigenous fauna including birds and reptiles has occurred in the absence of mammals for millions of years. This geo-biological situation means that New Zealand indigenous fauna and flora did not develop natural defence mechanisms against predation by mammals over time. For example, although many birds' sense of smell typically helps them to recognise the odour of potential predators (Amo et al., 2008; Röder et al., 2016), a study of the ability of exotic and indigenous avifauna in identifying mammals'scents in New Zealand suggests that indigenous species cannot respond to predator scent in an effective manner, and for this reason, they are more vulnerable to predator pressures from exotic fauna (Stanbury and Briskie, 2015).

1.3. The risk of predation in a New Zealand context

Evidence suggests that rising temperatures will increase the rate of predation upon indigenous fauna in New Zealand. According to the Ministry for the Environment (2014), the annual average temperature in the study area will increase by 0.9° C and 2.1° C, on average, by 2040 and 2090, respectively compared to 1990. For centuries, especially since the 1800s when Europeans came to New Zealand, indigenous biodiversity has been under a widespread biological attack across the country by invasive flora and fauna. Exotic mammals, in particular, are estimated to benefit from rising temperatures because rising temperatures lead to early flowering of flora, and the event of mast seeding phenomena which provide more food sources for these species. This, in turn, gives rise to a higher rate of reproduction among exotic mammals. This, consequently, increases the risk of increased predator pressures on indigenous fauna in New Zealand (Wilson et al., 1998; Pierce et al., 2006; McGlone and Walker, 2011; Christie, 2014). In addition, the outbreak and spread of diseases amongst indigenous fauna is very likely a result of an increase in the population of sources of infections triggered by rising temperatures (Baillie and Brunton, 2011; Ewen et al., 2012; Howe et al., 2012; Niebuhr, 2016).

The introduction of more than thirty exotic mammals to New Zealand, often known as pests, (Parkes and Murphy, 2003), among other causes, has led to widespread indigenous biodiversity loss across the country (McGlone, 1989; Towns and Daugherty, 1994; Saunders and Norton, 2001; Ewers et al., 2006). Common brushtail possum (*Trichosurus vulpecula*), for example, is an Australian mammal introduced to New Zealand in 1837 (Wodzicki, 1950). This species competes with indigenous fauna for food sources (Innes et al., 2010) and has a negative impact on the life-cycle of endemic trees (Cowan et al., 1997). European hedgehog (*Erinaceus europaeus*) predation affects populations of some endemic reptiles, insects, and ground-nesting birds (Jones et al., 2005; Reardon et al., 2012). Stoats (*Mustela erminea*) were introduced to New Zealand to control rabbit populations on farms (King, 2017). The stoats'rate of reproduction is very high and they predate largely upon indigenous avifauna (King, 1983). This can be related to the size of specie's home range (Murphy and Dowding, 1994). Weasels (*Mustela nivalis*) were introduced for the same purpose. This species is regarded as one of the main predators of indigenous avifauna and reptiles (King, 2017), and widely benefits from the overproduction of the seeds of beech trees during mast seeding events and this, accordingly, results in an increase in the number of individuals and therefore greater rates of predation. The aforementioned species are currently widespread in urban New Zealand. On the other hand, vegetation is highly fragmented in urban New Zealand due to a specific history of the urban development (Freeman and Buck, 2003; Meurk and Hall, 2006; Stewart et al., 2009; Rastandeh and Pedersen Zari, 2018). These conditions have cumulatively led to a higher level of vulnerability of indigenous fauna and flora.

1.4. Urban wildlife sanctuaries

Due to the fragile nature of New Zealand's indigenous fauna, in line with spatial management of land cover patterns at the class and landscape levels (Rastandeh and Pedersen Zari, 2018), a number of pest-free urban wildlife sanctuaries are needed in order to ensure the survival of vulnerable indigenous fauna against increased predator pressures and the spread of diseases triggered by rising temperatures in urban New Zealand. There is a wide range of evidence to suggest that the successful conservation of New Zealand indigenous fauna depends profoundly on pest control (Brown, 1997; Gillies and Clout, 2003; van Heezik et al., 2008; Innes et al., 2012; van Heezik et al., 2010; Aguilar et al., 2015; Goldson et al., 2015; Russell et al., 2015). Research on little-spotted kiwi (*Apteryx owenii*) (McLennan et al., 1996; Robertson and, Colbourne, 2004), other avifauna (Duncan and Blackburn, 2004; Blackburn et al., 2005; Innes et al., 2015a), tuatara (*Sphenodon*) (Jarvie

et al., 2016), and other reptiles (Reardon et al., 2012) indicates that the presence of introduced exotic mammals is a serious threat to the survival of the aforementioned species. Zealandia, the first pest-free urban wildlife sanctuary in New Zealand (Beatley, 2016), is a successful example of such strategies to respond to urban biodiversity loss. Zealandia was established in 1999 (Beatley, 2016) to support indigenous biodiversity in the Wellington urban landscape, but not necessarily as a refugium for indigenous fauna in the face of some impacts of climate change. This urban wildlife sanctuary, similar to other New Zealand examples (e.g. Travis Wetland Nature Heritage Park in Christchurch), has a special structure, a specific fence construction, a strategic visiting regime, and detailed management system that provide an isolated, pest-free, and accordingly disease-free, habitat for not only a wide range of vulnerable fauna, but indigenous flora in an urban context.

Although connectivity between patches of vegetation is essential to facilitate species movement and contribute to natural regeneration through seed dispersal and pollination mechanisms in urban New Zealand (Meurk and Hall, 2006; Meurk et al., 2016; Rastandeh and Pedersen Zari, 2018), pest-proof fencing is currently recommended by New Zealand experts for maintaining biodiversity in urban landscapes when biodiversity is to be addressed at the patch level (Burns et al., 2012; Innes et al., 2012; Empson and Fastier, 2013; Innes et al., 2015b; Norton et al., 2016). Although some of New Zealand's indigenous flying avifauna can still avail themselves of connectivity, highly vulnerable species such as little-spotted kiwi, tuatara, takahē (*Porphyrio hochstetteri*), hihi (*Notiomystis cincta*), and kākā (*Nestor meridionalis*),require active protection from predation pressures through the use of pest-proof fencing, or isolation on predator free offshore islands because they are not as mobile as New Zealand avifauna and/or their populations are not as large as other species. Therefore, site selection for suitable urban wildlife sanctuaries is vital to support the aforementioned vulnerable species in an era of climate change. Pest-free urban wildlife sanctuaries can simultaneously contribute to the presence and abundance of a wide range of other indigenous species endemic to New Zealand, accordingly.

2. Materials and methods

2.1. Study area

Wellington, New Zealand, a city with a population of more than 200,000 inhabitants, is one of the world's leading cities in urban biodiversity conservation (Clarkson and Kirby, 2016). The Wellington urban landscape is one of the most important hotspots for biodiversity conservation in urban New Zealand (Pedersen Zari, 2012, 2015; Rastandeh et al., 2017b). At the same time, the urban landscape of Wellington is highly fragmented. This area comprises of numerous patches of vegetation resulting from widespread urban development over the last two centuries (Rastandeh and Pedersen Zari, 2018). For this reason, it was selected as the case study area (Fig. 1).

2.2. Multi-criteria analysis

A six step process was designed, based upon Hill et al., 2005; Svoray et al., 2005; Wang and Hofe, 2008; Zucca et al., 2008; Huang et al., 2011; Fontana et al., 2013; Fernandez and Morales, 2015; Langemeyer et al., 2016,to perform a multi-criteria analysis of the most suitable patches of vegetation spatially capable of serving as urban wildlife sanctuaries in a New Zealand context under a climate that continues to becomewarmer. This is a cost-effective and relatively quick method to identify and rank candidate sites depending on expert knowledge, spatial data, and GIS analysis.

2.2.1. Step 1: identifying candidate sites

Approximately more than 60% of the Wellington urban landscape is covered by different types of green spaces ranging from indigenous,
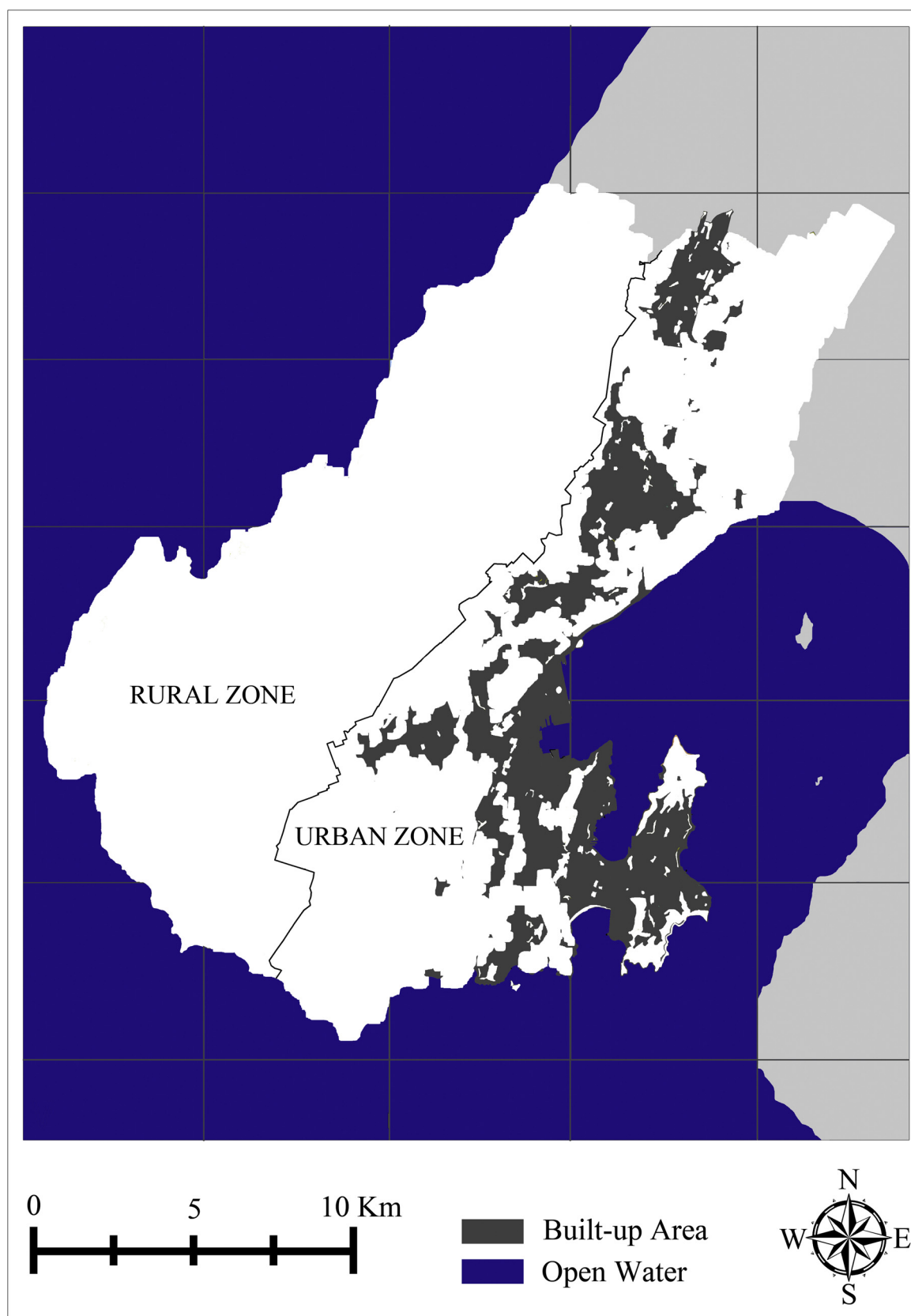
Fig. 1. Study area, the Wellington urban landscape (Urban Zone).

Table 1
Primary criteria for the identification of candidate sites.

Criteria	Rationale
Indigenousness	Areas covered by the class Indigenous Forest play an irreplaceable role in the provision of habitat and food for indigenous fauna in New Zealand (Rastandeh et al., 2017b; Table A1 in Appendix A).
Patch size	Larger patches of vegetation can support greater biodiversity over time. Based upon the current literature (e.g. Forman, 1995; Dramstad et al., 1996; Cornelis and Hermy, 2004) and discussions made in the course of interviews with six New Zealand subject-matter experts (i.e. Step 3), a threshold size of 25 ha was applied to identify the largest patches of vegetation.
Land ownership	The establishment of urban wildlife sanctuaries is more feasible in publically owned areas or in areas that are currently managed for biodiversity conservation.

Table 2
Total area and percentage of the candidate sites covered by the class Indigenous Forest.

Candidate site	Area (ha)	Indigenous Forest (ha)	% in the study area
Town Belt	98.8	22.27	18.32%
Botanic Garden	25.6	15.39	12.56%
Zealandia	247.6	28.94	23.74%
Otari-Wilton's Bush	81.2	16.71	13.72%
Total	453.2	83.31	68.34%

mixed, and exotic land cover classes. The rest is covered by built-up areas or non-vegetative patches (Rastandeh et al., 2017b). As green spaces differ in terms of habitat quality and ecological characteristics, it is necessary to focus on a limited number of candidate sites that are socially and/or ecologically suitable for this purpose. First, three primary criteria were taken into consideration to select the most relevant candidate sites (Table 1). Applying the primary criteria, four candidate sites were initially identified (Table 2). The selected candidate sites meet the primary criteria and encompass more than 68% of the class Indigenous Forest in the Wellington urban landscape (Table 2 & Fig. 2). From a biological point of view, the class Indigenous Forest is the most important land cover class in New Zealand, representative of New Zealand forests before the arrival of humans, and is of the highest habitat quality compared to other classes (Meurk and Hall, 2006; Landcare Research, 2015; Rastandeh et al., 2017b). Excluding Zealandia, the other three candidate sites are not currently fenced to protect vulnerable indigenous fauna and exclude pests. Land Cover Data Base v.4.1–LCDB (Landcare Research, 2015) and New Zealand Digital Elevation Model North Island 25 m – NZDEM (Landcare Research, 2010) were used to quantify the selected candidate sites in the Arc Map v.10.4.1 environment. FRAGSTATS v.4.2 (McGarigal et al., 2012) was also used to quantify and measure shape complexity based upon the metric Shape Index – SHAPE (Botequilha Leitão et al., 2006). As a size-independent algorithm, Shape Index is the most relevant landscape metric to quantify shape complexity (McGarigal et al., 2012).

2.2.2. Step 2: determining factors

Identification of factors characterising candidate sites, and thereby influencing indigenous fauna, was completed based upon the available published data. Current literature was systematically reviewed to identify the most important components of landscape patterns affecting the presence, abundance, and richness of fauna in urban landscapes (Rastandeh et al., 2017a). Eight components of landscape pattern are regarded as main factors for undertaking a suitability analysis: (1) indigenousness, (2) land cover heterogeneity, (3) land surface perviousness, (4) patch size, (5) connectivity and proximity, (6) edge density, (7) landform diversity, and (8) shape complexity (Table 3).

2.2.3. Step 3: dedicating weights to factors

To dedicate appropriate weights to each factor, expert-driven knowledge was sought. First, a questionnaire-centred survey of 87 international researchers with long-term track records of research on spatial planning for biodiversity conservation in urban landscapes was conducted to weight and rank the components in terms of importance

to biodiversity. More than 25% (n = 22) and 19% (n = 17) of participants mentioned that they have conducted at least one empirical research project in the Southern Hemisphere and Oceania, respectively. Participants were asked to dedicate a weight to each factor based upon their experience using the Likert Scale. They were also asked to consider the role of climate change as an important driver when weighting the factors. Next, six semi-structured interviews were conducted with subject-matter experts in New Zealand to adjust and confirm the results derived from the survey to ascertain that the derived weights are relevant in a New Zealand context (Table 3).

2.2.4. Step 4: defining attribute values and associated scores for each factor

Following Step 3 a suitability spectrum was designed in terms of factors identified in Step 2 in order to provide a framework for defining attribute values (Fig. 3). Possible attribute values for each factor were defined based upon current conditions using field surveys followed by a GIS analysis of the current composition and configuration of the four candidate sites. A set of rules was also defined to dedicate an individual score to each attribute value (Table 4).

2.2.5. Step 5: calculating weighted scores for each factor

The weighted score of each factor for each candidate site was calculated based upon the existing attribute of the candidate site using the following formula:

$$WS_i = W_i \times S_i$$

Where WS_i is the weighted score of factor i, W_i ($0 < W_i < 1$ and $\Sigma W_i = 1$) is the weight assigned to the factor i, and S_i is the score dedicated to the attribute value of factor i.

2.2.6. Step 6: calculating composite scores for each candidate site

Ultimately, the composite score of each candidate site was calculated using the following formula:

$$CS_i = \Sigma (WS_i)$$

Where CS_i is the composite score of factor i, and WS_i is the weighted score of factor i.

This six-step process was used to rank and prioritise identified candidate sites in terms of spatial suitability for safeguarding indigenous fauna in the face of increased predator pressures triggered by rising temperatures.

3. Results

Detailed information about the extent of land cover classes in the candidate sites has been provided in Tables 5 and 6. Zealandia, the Town Belt, Otari-Wilton's Bush, and Botanic Garden respectively cover 247.6 ha, 98.8 ha, 81.2 ha, and 25.6 ha of the study area (total area = 13511 ha) including a range of land cover classes of ecological significance. The largest patches of indigenous forest were identified in Zealandia (28.94 ha), Town Belt (22.27 ha), Otari-Wilton's Bush (16.71 ha), and Botanic Garden (15.39 ha). In terms of the ratio of the candidate site's area to indigenous forest area, Botanic Garden stands in first place because more than 60% of its total area is covered by the
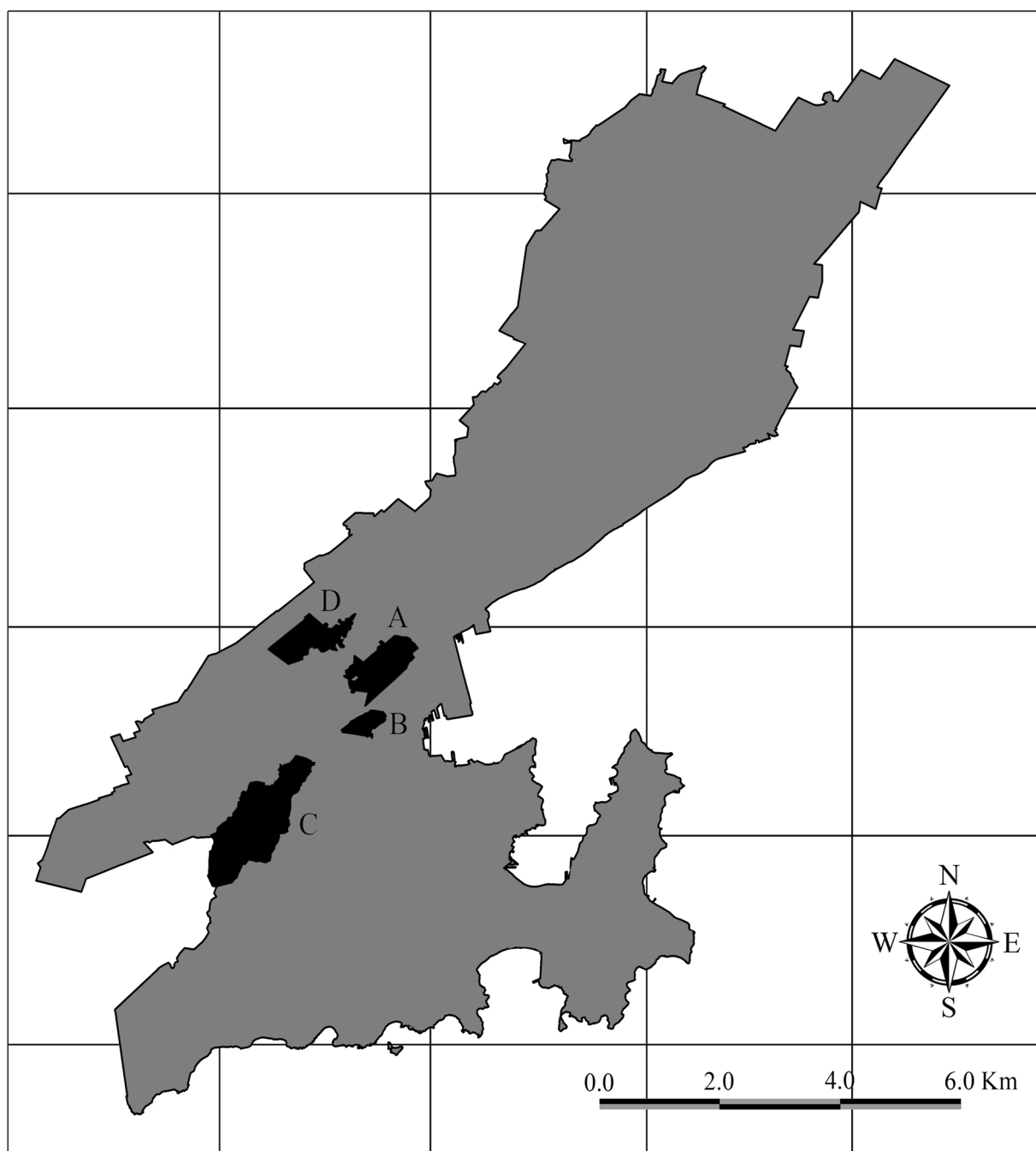
Fig. 2. The selected candidate sites in the Wellington urban landscape: Town Belt (A), Botanic Garden (B), Zealandia (C), and Otari-Wilton's Bush (D).

Table 3
Weights dedicated to the eight most important components of landscape pattern.

Factors	Definition	Weight
Indigenousness	The state of being indigenous/native in terms of land cover classes present in an urban context.	0.1377
Land cover heterogeneity	The diversity of different land cover classes in patch or landscape levels.	0.1449
Land surface perviousness	The ability of a particular land cover or landscape to absorb run-off caused by rainfall or sequester and store carbon dioxide in soil or vegetation.	0.1188
Patch size	Total area of a particular land cover on a patch or landscape level.	0.1639
Connectivity and proximity	The spatial distance between patches of a particular land cover class.	0.1449
Edge density	Perimeter of a patch of particular land cover class exposed to other land cover class.	0.0855
Landform diversity	The diversity of elevations, slopes, and aspects.	0.1188
Shape complexity	The degree to which a wildlife habitat is dissimilar to circle-shaped pattern.	0.0855

Highest proportion of indigenousness
Highest proportion of land cover heterogeneity
Highest proportion of land surface perviousness
As large as possible
As connected as possible
Lowest edge density
As topographically diverse as possible with large area covered by south-facing slopes
As compact as possible with lowest shape complexity

Most suitable attributes

Least suitable attributes

Lowest proportion of indigenousness
Lowest proportion of land cover heterogeneity
Lowest proportion of land surface perviousness
As small as possible
As disconnected as possible
Highest edge density
As topographically flat as possible with least area covered by south-facing slopes
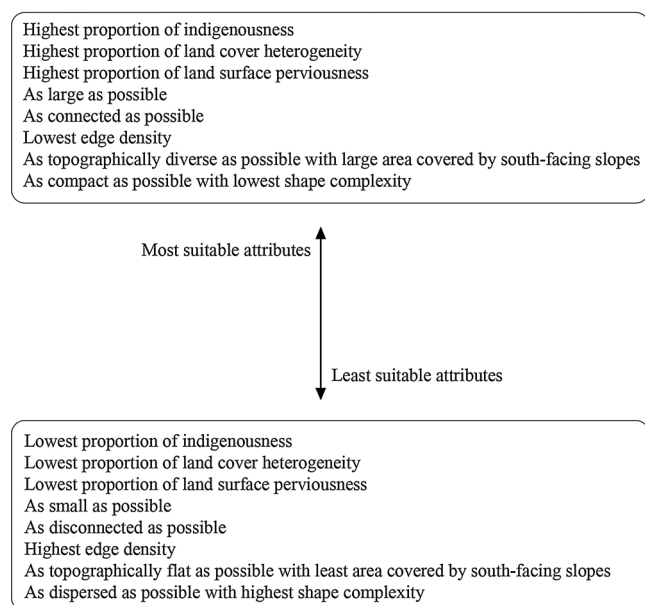As dispersed as possible with highest shape complexity

Fig. 3. Suitability spectrum used to define attribute values (cf. Table 4).

class Indigenous Forest. Detailed information about standard definitions of land cover classes as well as key flora species characterising each class is available in Table A1 in Appendix A.

Spatial attributes measured for the candidate sites in terms of the eight factors affecting wildlife species' behaviours in space and time were quantified and tabulated (Tables 7 and 8) using LCDB (Landcare Research, 2015), New Zealand Digital Elevation Model North Island 25 m (Landcare Research, 2010), and FRAGSTATS (McGarigal et al., 2012).

Zealandia is the most heterogeneous candidate site in terms of the number of land cover classes including the classes Built-up Areas and Lake or Pond (n = 7) (Fig. 4). In addition, the least pervious candidate site is Botanic Garden (land surface perviousness = 91.47%) compared to Zealandia (99.44%), Town Belt (98.62%), and Otari-Wilton's Bush (98.09%). Botanic Garden and Town Belt are the most connected candidate sites (255 m), while the spatial distance between Zealandia and the nearest patches of indigenous forest is 698m. The lowest edge density belongs to Botanic Garden (2671 m) whereas the values of edge density for other candidate sites are relatively high. Due to the lack of detailed spatial data, landform diversity cannot be easily measured on a fine scale in urban New Zealand. Despite this, a qualitative measurement of this factor followed by a GIS-based analysis of the candidate

sites using NZDEM shows that the highest landform diversity is available in Zealandia. Finally, in terms of shape complexity, a FRAGST-ATS-aided analysis of the candidate sites based on the landscape metric Shape Index (McGarigal et al., 2012) reveals that the most compact candidate sites are Botanic Garden (26.56), Otari-Wilton's Bush (33.23), Town Belt (42.17), and Zealandia (44.59), respectively. Ultimately, normalised and weighted composite scores of the candidate sites reveal that Zealandia (2.918) and Botanic Garden (2.769) are potentially the most important candidate sites in terms of spatial capability to help safeguard a range of vulnerable fauna against the local impacts of rising temperatures. Otari-Wilton's Bush (2.607) and Town Belt (2.325) stand in third and fourth places, respectively (Table 9).

4. Discussion

Zealandia and Botanic Garden are currently the most important sites that deserve to be managed for the conservation of indigenous biodiversity in the Wellington urban landscape, when urban biodiversity is addressed at the patch level and spatial isolation is required to avoid the spread of pests, diseases, and weeds.

From a spatial perspective, this investigation confirms that Zealandia, Wellington's first urban wildlife sanctuary, is indeed the most important site to help safeguard indigenous biodiversity against dispersal of pests and diseases as temperatures rise (Fig. 5). This site plays a key role in supporting indigenous biodiversity not only in the Wellington urban landscape, but at the national level because this urban wildlife sanctuary harbours a wide range of nationally vulnerable fauna including endangered and locally extinct species that bearly survive in the wild (Cote et al., 2013; Watts et al., 2014; Beatley, 2016; Shaw and MacKinlay, 2016; Nelson et al., 2016). Contrary to Botanic Garden, Otari-Wilton's Bush, and Town Belt, this site is currently fenced and is managed under effective predator control strategies.

Surprisingly, Botanic Garden, established in the 1840s (Shepherd and Cook, 1988), was identified as the second most important candidate site that is spatially capable of safeguarding vulnerable indigenous fauna (Fig. 6). This can be regarded as a significant finding because Botanic Garden has the lowest value in terms of patch size compared to other candidate sites. Although patch size was ranked as the most important component of landscape pattern by international researchers and by New Zealand subject-matter experts in Step 3, and this is in agreement with the current literature (e.g. Forman and Godron, 1981; Forman, 1995; Dramstad et al., 1996; Cornelis and Hermy, 2004; Kang et al., 2015; Sing et al., 2016), this finding suggests that patch size should not be considered as the single most important factor for the site selection of urban wildlife sanctuaries because the collective importance of other factors can outweigh the significance of the patch size itself.

Table 4

Scientific rules moderated based upon local conditions applied to dedicate scores to each factor's attribute.

Factors	Rules applied in the scoring system
Indigenousness	If 100% of the site is covered by the class Indigenous Forest, a score of 5 will be dedicated as the highest possible score. Other attributes will be relatively scored based upon this rule.
Land cover heterogeneity	If the number of land cover classes is 9, a score of 5 will be dedicated as the highest possible score. Other attributes will be relatively scored based upon this rule.
Land surface perviousness	If 100% of the site is covered by patches of vegetation, a score of 5 will be dedicated as the highest possible score. Other attributes will be relatively scored based upon this rule.
Patch size	A score of 5 will be dedicated as the highest possible score to the largest candidate site. Other attributes will be relatively scored based upon this rule.
Connectivity and proximity	If the spatial distance between the candidate site and the nearest patch covered by the class Indigenous Forest is less than 100m, a score of 5 will be dedicated as the highest possible score. Other attributes will be relatively scored based upon this rule.
Edge density	A score of 5 will be dedicated as the highest possible score to a candidate site with the lowest value of edge density. Other attributes will be relatively scored based upon this rule.
Landform diversity	A score of 5 will be dedicated as the highest possible score to a candidate site with the highest rate of topographical diversity. Other attributes will be relatively scored based upon this rule.
Shape complexity	Due to the lack of theoretically compact patches of vegetation (i.e. circle-shaped), a score of 3 will be dedicated as the relatively medium possible score to the most compact candidate site. Other attributes will be relatively scored based upon this rule.

Table 5

Land cover classes consisting of plant life in the candidate sites (cf. Fig. 4 and Table 6).

Candidate sites	Area (ha)	Indigenous classes	Exotic classes
Town Belt	98.8	Broadleaved Indigenous Hardwoods Indigenous Forest	Urban Parkland/Open Space Exotic Forest Gorse and/or Broom
Botanic Garden	25.6	Broadleaved Indigenous Hardwoods Indigenous Forest	Urban Parkland/Open Space
Zealandia	247.6	Broadleaved Indigenous Hardwoods Indigenous Forest Herbaceous Freshwater Vegetation	Exotic Forest Gorse and/or Broom
Otari-Wilton's Bush	81.2	Broadleaved Indigenous Hardwoods Indigenous Forest	Exotic Forest Gorse and/or Broom Low Producing Grassland

Table 6

Land cover classes identified in the candidate sites (cf. Fig. 4 and Table 5).

Land cover class	Town Belt (ha)	Botanic Garden (ha)	Zealandia (ha)	Otari-Wilton's Bush (ha)
Built-up Area	1.36	6.60	1.38	1.54
Indigenous classes				
Broadleaved Indigenous Hardwoods	38.37	1.43	166.05	62.41
Indigenous Forest	22.27	15.39	28.94	16.71
Herbaceous Freshwater Vegetation	0.00	0.00	0.41	0.00
Exotic classes				
Urban Parkland/Open Space	0.91	2.18	0.00	0.00
Gorse and/or Broom	1.41	0.00	1.11	0.39
Exotic Forest	34.50	0.00	46.04	0.08
Low Producing Grassland	0.00	0.00	0.00	0.14
Lake or Pond	0.00	0.00	3.73	0.00
Total	98.8	25.6	247.6	81.2

Table 8

Normalised scores calculated based upon rules defined in Step 5 (cf. Table 4).

Factors	Town Belt	Botanic Garden	Zealandia	Otari-Wilton's Bush
Indigenousness	1.127	3.004	0.584	1.027
Land cover heterogeneity	2.777	1.666	2.777	2.222
Land surface perviousness	4.931	4.573	4.972	4.904
Patch size	1.995	0.516	5	1.639
Connectivity and proximity	3	3	1.095	1.366
Edge density	2.201	5	1.444	1.601
Landform diversity	3	3	5	4
Shape complexity	1.889	3	1.786	2.397

Indigenous Forest through ecological succession processes. This is of critical significance because in a New Zealand context, ecological succession from the class Broadleaved Indigenous Hardwoods to the class Indigenous Forest has been observed (Davis and Meurk, 2001; Meurk and Hall, 2006; Williams, 2011; Wotton and McAlpine, 2013; q.v. Table A1 in Appendix A). Therefore, from a long-term planning perspective, Otari-Wilton's Bush and Town Belt are two candidate sites of great ecological significance to serve as urban wildlife sanctuaries in the face of future climate change, provided that they are managed appropriately over time. Similarly, 166.05 ha (67.06%) of Zealandia is currently covered by the class Broadleaved Indigenous Hardwoods. Appropriate management of natural regeneration mechanisms can considerably increase the expanse of the class Indigenous Forest in this site.

Although Botanic Garden is shown to be potentially a suitable site for harbouring indigenous fauna at the present time, the site is heavily visited by international and local visitors. The site is not fenced, yet. Therefore, uncontrolled access to the site by pests and people is currently possible. The presence of people on the site can affect fauna through producing light and noise pollution, and through feeding them (on purpose or not) which is associated with increased risk of disease (Jones and Reynolds, 2008). The presence of pets is also a problem to indigenous fauna when people visit this site. Thus, Botanic Garden can be considered as a socio-ecological system in an urban context that deserves specific spatially-explicit land cover pattern design and management in order to ensure the coexistence of people and wildlife species in the long run. This means that not all areas covered by Botanic

This research does not suggest that Otari-Wilton's Bush and Town Belt are not important for safeguarding vulnerable indigenous fauna. The ranking presented in this research is completely relative. It means that the importance of each candidate site has been defined in relation to the other three. The methodology undertaken helps to establish a scientific-based platform for the identification, ranking and prioritisation of candidate sites for urban wildlife sanctuaries in terms of current compositions and configurations of patches of vegetation in a highly fragmented urban landscape. The suggested process helps planners and urban leaders to make appropriate decisions regarding the establishment of new pest-free urban wildlife sanctuaries, which as discussed are needed as the impacts of climate change become more acute. The ecological importance of Otari-Wilton's Bush and Town Belt is confirmed through the process, because analysis shows that 62.41 ha (76.75%) and 38.37 ha (38.83%) of Otari-Wilton's Bush and Town Belt respectively are covered by the class Broadleaved Indigenous Hardwoods, while only 1.43 ha (5.58%) of Botanic Garden is covered by this class currently (Table 6). If managed appropriately, the relatively large patches of the class Broadleaved Indigenous Hardwoods in Otari-Wilton's Bush and Town Belt can be replaced by the class

Table 7

Attributes measured for the candidate sites in terms of the eight factors characterising the spatial composition and configuration of the candidate sites.

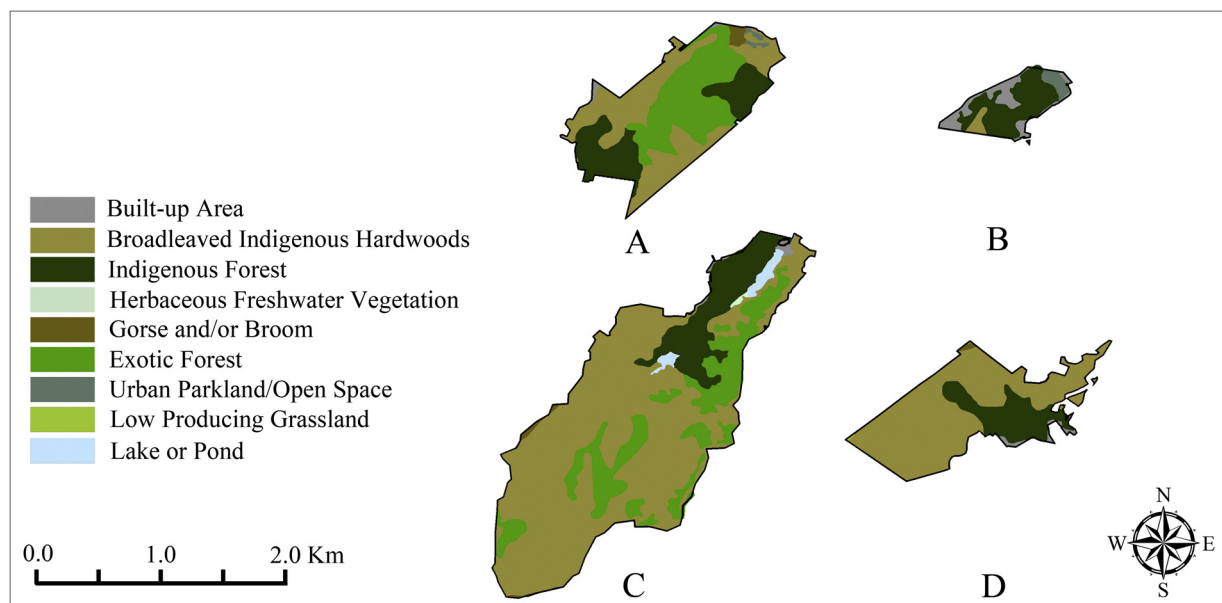| Factors	Town Belt	Botanic Garden	Zealandia	Otari-Wilton's Bush
Indigenousness	22.3 ha (22.54%)	15.4 ha (60.09%)	28.9 ha (11.69%)	16.7 ha (20.55%)
Land cover heterogeneity (with reference to Table 4)	5	3	5	4
Land surface perviousness	98.62%	91.47%	99.44%	98.09%
Patch size	98.8 ha	25.6 ha	247.6 ha	81.2 ha
Connectivity and proximity	255m	255m	698m	560m
Edge density	6066m	2671m	9243m	8339m
Landform diversity	Moderate	Moderate	Very high	High
Shape complexity	42.17	26.56	44.59	33.23

Fig. 4. Land cover classes in the candidate sites: Town Belt (A), Botanic Garden (B), Zealandia (C), and Otari-Wilton's Bush (D).

Table 9
Normalised and weighted composite scores dedicated to the candidate sites in Step 6.

Factors	Town Belt	Botanic Garden	Zealandia	Otari-Wilton's Bush
Indigenousness	0.155	0.413	0.080	0.141
Land cover heterogeneity	0.402	0.241	0.402	0.321
Land surface perviousness	0.585	0.543	0.590	0.582
Patch size	0.326	0.084	0.819	0.268
Connectivity and proximity	0.434	0.434	0.158	0.198
Edge density	0.188	0.442	0.123	0.136
Landform diversity	0.356	0.356	0.594	0.475
Shape complexity	0.161	0.256	0.152	0.204
Normalised and weighted composite scores	2.607	2.769	2.918	2.325

Garden are recommended for fencing, but rather, areas of social and ecological importance should be separately managed in a way that minimises the negative impacts on indigenous biodiversity. The latter case, requires further research on this site.

The methodology designed and applied in this research highlights four key implications for landscape architecture and spatial planning in urban landscapes. First, multi-criteria analysis of landscape pattern composition and configuration of patches of vegetation is a scientific-based, cost-effective, and relatively quick method to provide urban decision makers with a detailed spatial picture of opportunities for identifying and ranking locations as potential urban wildlife

sanctuaries, in order to help safeguard vulnerable indigenous fauna in urban landscapes. This is required to mitigate negative impacts imposed by rising temperatures such as increased predator pressures and the outbreak of diseases among vulnerable indigenous fauna.

Second, although very large patch size is thought by many to be a panacea for safeguarding biodiversity from climate change impacts in urban landscapes, this research indicates that patch size should not be regarded as the single most important factor in the process of site selection, because even small patches of vegetation may provide indigenous fauna with spatially suitable habitats in the face of some impacts imposed by climate change. This is true when habitat quality is met through other factors and collective importance of other factors outweighs the role of patch size. The relative location of patches in relation to each is also a factor particularly when considering flying bird species. Despite the fact that this research is based upon a small sample of four candidate sites in a New Zealand context, it can be considered a valuable area of inquiry for future research in other contexts in terms of ecologies, climates, and social settings.

Third, it is particularly important to recognise the significance of some patches of vegetation that are capable of ecological succession through landscape restoration practices and/or natural regeneration mechanisms over time. Therefore, while the importance of patches of indigenous forest should not be overestimated for the site selection of urban wildlife sanctuaries, the importance of understanding the potential long-term role of some plant and bird species in increasing the percentage of indigenousness in urban landscapes through the aforementioned processes should not be neglected.

Fig. 5. Expanses of indigenous flora associated with land cover heterogeneity and landform diversity in Zealandia.

Fig. 6. Expanses of indigenous flora associated with land cover heterogeneity and landform diversity in Botanic Garden.

Fourth, the presence of people in designed and semi-natural urban green spaces is inevitable, and from a psychological wellbeing point of view is to be encouraged (Pedersen Zari, 2017). Where the rate of human-wildlife conflict of interests is high, a middle ground between societal demands and biodiversity requirements should be sought to minimise anthropogenic impacts on indigenous fauna, and concurrently respect cultural and social values.

5. Conclusion

Biodiversity loss is increasing in urban landscapes. Biological invasion by exotic species triggered by rising temperatures is an issue of critical importance in urban New Zealand suffering from rapid urban growth and consequently landscape fragmentation since the 1800s. New Zealand indigenous fauna are among taxonomic groups that are highly vulnerable to predator pressures. In line with informed spatial planning for biodiversity conservation across urban landscapes (Rastandeh and Pedersen Zari, 2018), decision makers in urban New Zealand need to recognise opportunities for safeguarding vulnerable indigenous fauna against predation, including informed decision-making on when and where isolated pest-free urban wildlife sanctuaries can be established to help avoid the spread of pests, diseases, and weeds. As shown spatially, there are opportunities for establishing new pest-free urban wildlife sanctuaries in the Wellington urban landscape. Despite widespread land cover change and dense human activities in the region, this research reveals where, how, and why urban wildlife sanctuaries should be established in response to increased predator pressures and the spread of diseases among vulnerable indigenous fauna triggered by rising temperatures. The methodology employed in this research can be transferable to other New Zealand urban landscapes.

Appendix A

Table A1
Land cover classes in the candidate sites.

Land cover class	Definition (Landcare Research, 2015)
Built-up Area (settlement)	Commercial, industrial or residential buildings, including associated infrastructure and amenities, not resolvable as other classes. Low density 'lifestyle' residential areas are included where hard surfaces, landscaping and gardens dominate other land covers.
Urban Parkland/Open Space	Open, mainly grassed or sparsely-treed, amenity, utility and recreation areas. The class includes parks and playing fields, public gardens, cemeteries, golf courses, berms and other vegetated areas usually within or associated with built-up areas.
Lake or Pond	Essentially-permanent, open, fresh-water without emerging vegetation including artificial features such as oxidation ponds, amenity, farm and fire ponds and reservoirs as well as natural lakes, ponds and tarns.
Low Producing Grassland	Exotic sward grassland and indigenous short tussock grassland of poor pastoral quality reflecting lower soil fertility and extensive grazing management or non-agricultural use. Browntop, sweet vernal, danthonia, fescue and Yorkshire fog dominate, with indigenous short tussocks (hard tussock, blue tussock and silver tussock) common in the eastern South Island and locally elsewhere.
Herbaceous Freshwater Vegetation	Herbaceous wetland communities occurring in freshwater habitats where the water table is above or just below the substrate surface for most of the year. The class includes rush, sedge, restiad, and sphagnum communities and other wetland species, but not flax nor willows which are mapped as Flaxland and Deciduous Hardwoods respectively.
Gorse and/or Broom	Scrub communities dominated by gorse or Scotch broom generally occurring on sites of low fertility, often with a history of fire, and insufficient grazing pressure to control spread. Left undisturbed, this class can be transitional to Broadleaved Indigenous Hardwoods.
Broadleaved Indigenous Hardwoods	Lowland scrub communities dominated by indigenous mixed broadleaved shrubs such as wineberry, mahoe, five-finger, Pittosporum spp, fuchsia, tutu, titoki and tree ferns. This class is usually indicative of advanced succession toward indigenous forest.
Indigenous Forest	Tall forest dominated by indigenous conifer, broadleaved or beech species.
Exotic Forest	Planted or naturalised forest predominantly of radiata pine but including other pine species, Douglas fir, cypress, larch, acacia and eucalypts. Production forestry is the main land use in this class with minor areas devoted to mass-movement erosion-control and other areas of naturalised (wildling) establishment.

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